

Modifying FHANTM 2.0 to Estimate Phosphorus Concentrations in Runoff from Mid-Atlantic Coastal Plain Soils

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ABSTRACT

Phosphorus lost from agricultural soils has been identified as a nonpoint source pollutant of surface waters in Delaware and throughout the Mid-Atlantic Coastal Plain. The Field Hydrologic and Nutrient Transport Model (FHANTM) 2.0 can help identify areas with a high potential for P loss, but the method used to estimate P concentrations in runoff waters needs reevaluation. The equation $P_d = KP_o^a W^b$ has been proposed to predict P desorption from soil to runoff. To test this equation for use in Delaware and the Mid-Atlantic Coastal Plain, we conducted rainfall simulations for 14 Delaware and Maryland soils packed into 5 by 20 by 100 cm boxes at a rainfall intensity of 7.5 cm h⁻¹ and a slope of 5% for 30 min. We collected all runoff and measured an average soluble P concentration in runoff for the entire simulation. We predicted P concentrations using the above equation and compared them with measured values. Predicted values were well correlated to measured values ($r^2 = 0.78$), but P concentrations were overpredicted by an average of 20 times. After we added a calibration factor to the equation based on the amount of sediment lost in runoff during the rainfall simulation, measured and predicted soluble P concentrations exhibited a nearly 1:1 relationship. Results suggest that eroded sediment in runoff may resorb P from the runoff solution, causing the desorption equation to overpredict soluble P concentrations in runoff.

BECAUSE OF THE CONCERN that the movement of P from agricultural soils to surface waters contributes to eutrophication, the last several years have seen a great deal of research conducted nationally and internationally to develop methods to quantify the risk of P loss from agricultural areas. In the state of Delaware, water quality in the Inland Bays national estuary has been impaired by P. To comply with the Clean Water Act (United States, 1967), Total Maximum Daily Loads (TMDLs) have been established for the Bays. The goal of the TMDLs for P is a 70% reduction of nonpoint source P loads coming into the Bays (Delaware Department of Natural Resources and Environmental Control [DNREC, 1998]). Given this, Delaware has become one of the many places where research is being conducted to both identify agricultural areas that have a high potential for P export and to develop methods to quantify that export.

Nutrient transport models are one way to estimate the risk of agricultural P degrading water quality. In Florida, FHANTM 2.0 (Fraisie and Campbell, 1997)

was developed to simulate field-scale water and P movement as part of an effort to reduce P loads to Lake Okeechobee. FHANTM 2.0's hydrology is based on DRAINMOD (Skaggs, 1980), and its nutrient components are based on GLEAMS (Leonard et al., 1987). Because Florida and Delaware have the physical and hydrologic conditions of flat fields, high water tables, and drainage hydrology, FHANTM 2.0 can potentially be used in Delaware to simulate field-scale P export. However, some of FHANTM 2.0's simulations of soil P processes were designed either for pesticide chemistry in soil or for specific chemical and physical properties of Florida soils that are not applicable to Delaware soils. These P processes should be modified to more accurately represent Delaware soils. One such modification is the prediction of soluble P concentrations in runoff waters.

Currently in FHANTM 2.0, the concentration of P in runoff, $(C_w)_p$ (mg L⁻¹), is calculated based on partitioning (K_d) and extraction (B) coefficients as:

$$(C_w)_p = [(C_{av})_p B] / [1 + (K_d) B] \quad [1]$$

where $(C_{av})_p$ (mg kg⁻¹) is the quantity of P in the topsoil available for runoff. Because Eq. [1] was originally designed to represent pesticide transformations in soils (Leonard et al., 1987) and was calibrated in FHANTM 2.0 for Florida soils, it may not accurately predict soluble P concentrations in runoff from Delaware soils. For example, K_d values for the A horizons in FHANTM 2.0 are calculated based on soil Mg content. Research for Delaware soils has not shown any relationship between soil Mg and soil P adsorption, desorption, or transformations (Vadas, 2001). In fact, soil Mg data is largely unavailable for Delaware soils. Therefore, an effort to see if the current algorithms of FHANTM 2.0 may indeed work for Delaware soils is in some respects not even possible.

Because it has commonly been observed that P desorption from soil to water occurs rapidly at first and then decreases as equilibrium is approached, and that the quantity of P desorbed is largely a function of de-

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Abbreviations: Al_{ox}, acid ammonium oxalate-extractable Al; B , extraction coefficient used in FHANTM 2.0; $(C_{av})_p$, quantity of P in the topsoil available for runoff used in FHANTM 2.0; $(C_w)_p$, concentration of P in runoff used in FHANTM 2.0; EDI, effective depth of interaction; Fe_{ox}, acid ammonium oxalate-extractable Fe; FHANTM 2.0, field hydrologic and nutrient transport model, Version 2.0; ICP-AES, inductively coupled plasma atomic emission spectroscopy; K , empirical constant in P desorption equation; K_d , partitioning coefficient used in the FHANTM 2.0 model; OC, organic C; P_o , initial concentration of desorbable P in P desorption equation; PL, poultry litter; STP, soil test P; t , Time (min) of P desorption in P desorption equation; TMDL, total maximum daily load; UDSTP, University of Delaware Soil Testing Program; W , water/soil ratio (cm³ g⁻¹) during P desorption in P desorption equation; α , empirical constant in P desorption equation; β , empirical constant in P desorption equation.

sorption time and the desorption water/soil ratio (Barrow, 1979), Sharpley et al. (1981a) proposed a P desorption equation of the form:

$$P_d = K P_o t^\alpha W^\beta \quad [2]$$

where P_d is the amount of P desorbed (mg kg^{-1}) in time t (min) at a water/soil ratio W ($\text{cm}^3 \text{g}^{-1}$), P_o is the initial amount of desorbable P present in the soil (mg kg^{-1}), and K , α , and β are empirical constants for a given soil. In a previous paper that developed the intent to incorporate Eq. [2] into FHANTM 2.0 for use in Delaware, Vadas and Sims (2002) proposed equations to predict K , α , and β from easily determined soil properties as:

$$\alpha = 0.024 (\text{Fe}_{\text{ox}}/\text{OC})^{-0.346} \quad [3]$$

$$\beta = 0.107 (\text{Fe}_{\text{ox}} + \text{Al}_{\text{ox}})^{0.251} \quad [4]$$

$$K = 2.168 (\text{Fe}_{\text{ox}} + \text{Al}_{\text{ox}})^{-0.773} \quad [5]$$

where Fe_{ox} and Al_{ox} are acid ammonium oxalate-extractable Fe (mmol kg^{-1}) and Al (mmol kg^{-1}), and OC is organic C (mmol kg^{-1}). The objective of the present research was to determine if Eq. [2] through [5] can accurately predict soluble P concentrations in runoff using simulated rainfall studies that attempt to represent natural runoff conditions better than the laboratory experiments used to generate Eq. [3] through [5]. Even though this research was conducted with a focus on Delaware soils, it should be applicable for similar soils throughout the Mid-Atlantic Coastal Plain.

MATERIALS AND METHODS

Soil Selection and Characterization

Seven agricultural topsoils (~0–20 cm) were collected from sites in Delaware and Maryland. The Delaware soil types included a Matapeake silt loam (fine-silty, mixed mesic Typic Hapludult) and a Butlerstown silt loam (fine-silty, mixed semi-active mesic Typic Fragiudult) from the coastal plain region in northern Delaware, and an Evesboro loamy sand (mesic, coated, Typic Quartzipsamment), a Sassafras loamy sand (fine loamy, siliceous, mesic Typic Hapludult), and a Pocomoke loamy sand (coarse-loamy, siliceous, thermic Typic Umbracquilt) from the coastal plain region in southern Delaware. Each soil was collected from four different locations at the same site to give four different levels of soil P within the same soil type. The two Maryland soils also included a Matapeake silt loam and an Evesboro sandy loam from the coastal plain region of Maryland's Eastern Shore. Similarly, these Matapeake and Evesboro soils were each collected from three locations at the same site to give three different levels of soil P within the same soil type. In total, there were 26 different soil/soil P level combinations. The different P levels within the same soil type were a result of varying management practices at a given site, especially manure application history. All soils were air-dried and sieved to 7 mm.

Simulated Rainfall Experiments

Soils were packed in triplicate into wooden runoff boxes that were 100 cm long by 20 cm wide by 7.5 cm deep and had impermeable bottoms. Soil box design was in accordance with the National P Research Project (North Carolina State University, 2002). Soil in boxes was leveled to a depth of 5 cm, and

densities averaged about 1.7 g cm^{-3} for the silt loams and about 1.4 g cm^{-3} for the sandy loams and loamy sands. Each box was treated as a discrete individual, and runoff results were not averaged for each set of triplicates. In total, there were 78 individual boxes subjected to simulated rainfall. Twenty-four hours before being placed under a rainfall simulator, soils in the boxes were prewetted with simulated rain to the point that runoff visually appeared about to begin. Soils were prewetted so that runoff would start as immediately and as uniformly as possible for all boxes once simulated rainfall began. The following day, just prior to rainfall simulation, soil samples were taken from the runoff boxes at the entire soil depth (5 cm). Runoff does not react with all soil to 5 cm, but because the soil in the boxes was homogenized, these samples represented the soil that would interact with runoff. It was not determined if the prewetting caused any significant downward movement of P so that the soil at the surface in the boxes contained less P than the soil deeper in the boxes. Holes left from the removed soil samples were replaced with the respective soil of that box. Soil samples were dried and sieved to 2 mm and characterized for pH [1:1 soil/water ratio], soil test P (STP; Mehlich-1 extraction; 1:4 ratio of soil/0.05 M HCl + 0.0125 M H_2SO_4 ; 5-min reaction time [Sims and Heckendorn, 1991]), particle size by the hydrometer method (Bouyoucos, 1962), acid ammonium oxalate-extractable Al and Fe [Al_{ox} and Fe_{ox} ; 1:40 ratio of soil/0.2 M $(\text{NH}_4)_2\text{C}_2\text{O}_4$; 2-h reaction time in darkness (McKeague and Day, 1966)], OC by the Walkley-Black wet oxidation procedure (Nelson and Sommers, 1982), and the initial amount of desorbable P (P_o , Eq. [2]) with Fe oxide impregnated filter strips [1:40 ratio of soil/0.01 M CaCl_2 + Fe oxide coated filter paper strip; 16-h reaction time, followed by desorption of P with 1 M H_2SO_4 (Chardon et al., 1996)]. Vadas and Sims (2002) explain why the Fe oxide strip method was a reasonable estimate of soil desorbable P. The P in the Mehlich-1 extraction and the Fe and Al in the oxalate extraction were measured by inductively coupled plasma atomic emission spectroscopy (ICP-AES). The P in the filter strip procedure was measured by the molybdate blue method of Murphy and Riley (1962) with absorbance measured at 882 nm.

The rainfall simulator was based on the design of Miller (1987), with one TeeJet $\frac{1}{2}$ HH-SS50WSQ nozzle (Spraying Systems Co., Wheaton, IL) placed in the center of the simulator and ~3 m above the soil surface in the boxes. The nozzle and associated water piping, pressure gauge, and electrical wiring were mounted on an aluminum frame that was ~3.3-m square and ~3.4 m high. A pressure regulator was used to establish a water-flow rate of about 120 mL s^{-1} at the nozzle. A rainfall intensity of 70 mm h^{-1} was achieved by controlling the relative on/off spraying times of the nozzle with a solenoid valve until the desired intensity was attained. Rainfall intensity and uniformity were determined by collecting samples in 144 cups (250 mL) spaced on a uniform grid covering the entire area under the simulator. Intensity was determined by weighing the amount of water in each cup. Uniformity was determined by taking an average weight of water for all 144 cups, determining the difference between the water weight and this average for each cup, averaging all these differences, dividing the average difference by the average of water weights, and then subtracting this final number from 1 to calculate uniformity. Uniformity was consistently greater than 80%.

Six soil boxes at a time were placed under the rainfall simulator at a slope of 5%, and rainfall experiments were conducted at an intensity of 75 mm h^{-1} for 30 min. These conditions are in accordance with the National P Research Project (North Carolina State University, 2002). Runoff began almost immediately following the onset of rainfall, and runoff came

from the entire surface area of the box and was not concentrated at the front of the box near the outlet. For each box, all runoff was collected in a single container. Each container was weighed to determine the total volume of runoff assuming that 1 kg equaled 1 L of runoff. The relatively low sediment content in the runoff justified this assumption. The runoff containers were immediately transferred to the laboratory to analyze the runoff. Each container was placed on a large stir plate to generate constant mixing. One 50-mL subsample was extracted with a plastic syringe to determine sediment concentration by evaporation in an oven. A second 50-mL subsample was taken to determine soluble P concentration. The sample was filtered through a 0.45- μm filter paper and analyzed for soluble P colorimetrically on a Sequoia-Turner model 340 spectrophotometer (Sequoia-Turner Corp., Mountain View, CA) by the molybdate blue method of Murphy and Riley (1962) with absorbance measured at 882 nm.

Calculations

Equation [2] was used to predict the amount of P desorbed from the soil in the boxes to the collected runoff water during the simulated rainfall. Values for K , α , and β were calculated using Eq. [3] through [5]. The P_0 (mg kg^{-1}) was measured with Fe oxide strips before rainfall as described above. Time, t (min), was 30 min for the entire period of the runoff event. The water/soil ratio, W ($\text{cm}^3 \text{g}^{-1}$), was calculated as the volume of runoff (cm^3) divided by the mass of soil (g) that interacted with the runoff with the equation:

$$W (\text{cm}^3 \text{g}^{-1}) = \frac{\text{Runoff (mm)}}{[\text{EDI (mm)}] [D_b (\text{g cm}^{-3})]} \quad [6]$$

where EDI is the effective depth of interaction and D_b is the soil bulk density. The EDI was calculated based on the equation of Sharpley (1985):

$$\ln \text{EDI} = (i) + ((0.576) [\ln [\text{soil loss (kg ha}^{-1})]]) \quad [7]$$

where i is related to the degree of soil aggregation by the equation of Sharpley (1985):

$$i = (-3.130) + [(0.071) (\text{soil aggregation})] \quad [8]$$

where soil aggregation is the proportion of clay-sized material ($<2 \mu\text{m}$) measured during particle-size analysis in dispersed and undispersed soil samples. The degree of soil aggregation was not measured for these soils, so the soils were given values as measured by Sharpley (1985) for similar soil types. The values were 26 for the silt loam soils and 20 for the sandy loam soils. The D_b values averaged 1.7 g cm^{-3} for the silt loams and 1.4 g cm^{-3} for the sandy loams and loamy sands as determined by the amount of soil packed into the boxes. Soil loss for Eq. [7] was measured in runoff as grams per liter and was converted to kilogram per hectare with the equation:

$$\text{Soil Loss (kg ha}^{-1}) =$$

$$\frac{[\text{Soil Loss (g L}^{-1})] \times [\text{Runoff (L)}] \times 100^4 (\text{cm}^2 \text{ha}^{-1})}{1000 (\text{g kg}^{-1}) \times 2000 (\text{cm}^2 \text{box}^{-1})} \quad [9]$$

The units of runoff in Eq. [6] were converted from liters to millimeters with the equation:

$$\text{Runoff (mm)} = \frac{[\text{Runoff (L)}] [1000 (\text{cm}^3 \text{L}^{-1})] [10 (\text{mm cm}^{-1})]}{2000 (\text{cm}^2 \text{box}^{-1})} \quad [10]$$

Equation [2] predicts the amount of P that desorbs from the soil, P_d , in milligrams per kilogram (mg kg^{-1}). To convert this value to milligrams per liter to predict the concentration of soluble P in runoff, it must be multiplied by a term with the units kilograms per liter, with kilograms being the mass of soil that interacted with runoff and liter being the volume of runoff (Sharpley et al., 1981a; Sharpley et al., 1985). To make this conversion, the units for runoff were first converted from liters to millimeters using Eq. [10], and the conversion factor of kilograms per liter was calculated by the equation:

$$(\text{kg L}^{-1}) = \frac{[\text{EDI (mm)}] [D_b (\text{g cm}^{-3})]}{\text{Runoff (mm)}} \quad [11]$$

The final units for the right side of Eq. [11] are actually grams per cubic centimeter (g cm^{-3}), but these units were assumed to be equivalent to kilogram per liter (kg L^{-1}). The calculation in Eq. [11] is the same as dividing by W in Eq. [6]. These units calculations above were made to be consistent with the units used in FHANTM 2.0. The above series of equations should also be useful more generally to researchers who want to compare runoff or P desorption results from natural or simulated rainfall/runoff studies to the predictions of other mathematical models that use similar units.

RESULTS AND DISCUSSION

Soil and Runoff Characteristics

The soils used in this study were representative of many soils in the Mid-Atlantic Coastal Plain. Soils were generally sandy or silty, moderately acidic, and low in OM, although this varied slightly (Table 1). Mehlich-1 STP values ranged widely from 26 to 394 mg kg^{-1} , with the average value of 170 mg kg^{-1} . The agronomic STP rating system used by the UDSTP ranks soils as low ($<13 \text{ mg P kg}^{-1}$), medium ($13\text{--}24 \text{ mg P kg}^{-1}$), optimum ($25\text{--}50 \text{ mg P kg}^{-1}$), or excessive ($>50 \text{ mg P kg}^{-1}$).

The volume of runoff was similar for all soil boxes, ranging from 4.2 to 7.0 L, with an average of 5.5 L. This is in comparison with the intended volume of 7.0 L of rainfall that was set to actually fall on each soil box in

Table 1. Selected soil physical and chemical properties of the 78 soils from soil boxes just before the simulated rainfall experiments.

Property	All soils	Maryland Soils	Delaware Soils
		Mean ± SD†	
pH	5.8 ± 0.3	5.9 ± 0.2	5.7 ± 0.4
Sand, %	50 ± 29	47 ± 28	52 ± 29
Clay, %	16 ± 6	17 ± 6	16 ± 6
OM, %	2.3 ± 1.7	1.7 ± 0.7	2.5 ± 1.8
Oxalate Fe, mg kg ⁻¹	734 ± 572	1356 ± 750	524 ± 270
Oxalate Al, mg kg ⁻¹	1095 ± 896	1434 ± 747	1016 ± 876
Mehlich 1-P, mg kg ⁻¹	128 ± 98	55 ± 142	124 ± 108
Fe-oxide strip labile P, mg kg ⁻¹	23 ± 21	8 ± 26	24 ± 2

† Mean \pm standard deviation.

30 min. The failure to always collect 7.0 L of runoff was likely because of variability in soil retention of water and in rainfall distribution. Sediment concentration in runoff varied more than runoff volume and was closely related to soil texture. For silt loams, sediment concentration ranged from 1.5 to 7.0 g L⁻¹, with an average of 4.0 g L⁻¹. For sandy soils, sediment concentration ranged from 0.3 to 2.8 g L⁻¹, with an average of 1.4 g L⁻¹.

Predicting Soluble Phosphorus in Runoff

Equations [2] through [11] were used to predict soluble P concentrations in runoff. Figure 1 shows that measured soluble P concentrations in runoff ranged from 0.001 to 0.418 mg P L⁻¹, and that Eq. [2] through [11] overpredicted soluble P in runoff by a factor of about 23. Sharpley and Smith (1989) used a similar series of equations to predict soluble P in runoff from natural rainfall for 20 agricultural watersheds in Oklahoma and Texas that were mostly in grassland, with some cultivated crops or a mix of the two. They accurately predicted soluble P concentrations in runoff when runoff was >75 mm, but overpredicted P concentrations by a factor of two to 12 when runoff was <75 mm. They attributed their overprediction errors to using a constant value for EDI of 0.3 mm. When EDI was allowed to fluctuate according to soil loss (Eq. [7]), predicted values of soluble P in runoff were not significantly different from measured values. Apparently, inappropriate values for EDI caused inappropriate values for W (Eq. [6]) and for soluble P concentrations (Eq. [11]). Overall, Sharpley and Smith (1989) stressed that accurate representation of the interaction of soil and runoff, as expressed by W and EDI, is necessary for predicting soluble P loss in runoff.

Daniel et al. (1993) used equations similar to Eq. [2] through [11] to predict soluble P in runoff from simulated rainfall on 38 field plots with treatments of pasture, tillage with residue, and tillage without residue. They found that predicted values of soluble P in runoff were similar to measured values for pasture plots, but were greater than measured values by a factor of about 20 for tillage plots. This was in spite of accounting for variations in EDI, and thus W, using Eq. [7]. As well, measured soluble P in runoff from pasture plots was approximately five times greater than soluble P in runoff from tillage plots, even though all plots had the same soil type, and thus values for K, α , and β (Eq. [3]–[5]), and a similar range in soil P concentrations. The authors suggested that runoff P was greater for pasture plots because of P leaching from turf residue, dilution of runoff P from the greater runoff amounts from tillage plots, and resorption of P by suspended sediment in tillage plot runoff. Average runoff was five times greater from tillage plots than from pasture plots, and average sediment loss was three orders of magnitude greater from tillage plots than from pasture plots.

Our results and the results of Sharpley and Smith (1989) and Daniel et al. (1993) suggest that Eq. [2] through [11] can predict soluble P in runoff well for pasture systems, but are likely to overpredict soluble P in runoff for cultivated systems where soil loss in runoff is much greater. An examination of Eq. [2] through [11] shows several possible sources that may have caused overprediction errors in our experiments. First is estimating values for K, α , and β (Eq. [2] through [5]) and estimating P₀ (Eq. [2]). The work by Vadas and Sims (2002) provides justification for the methods used to estimate these parameters, suggesting that they are not

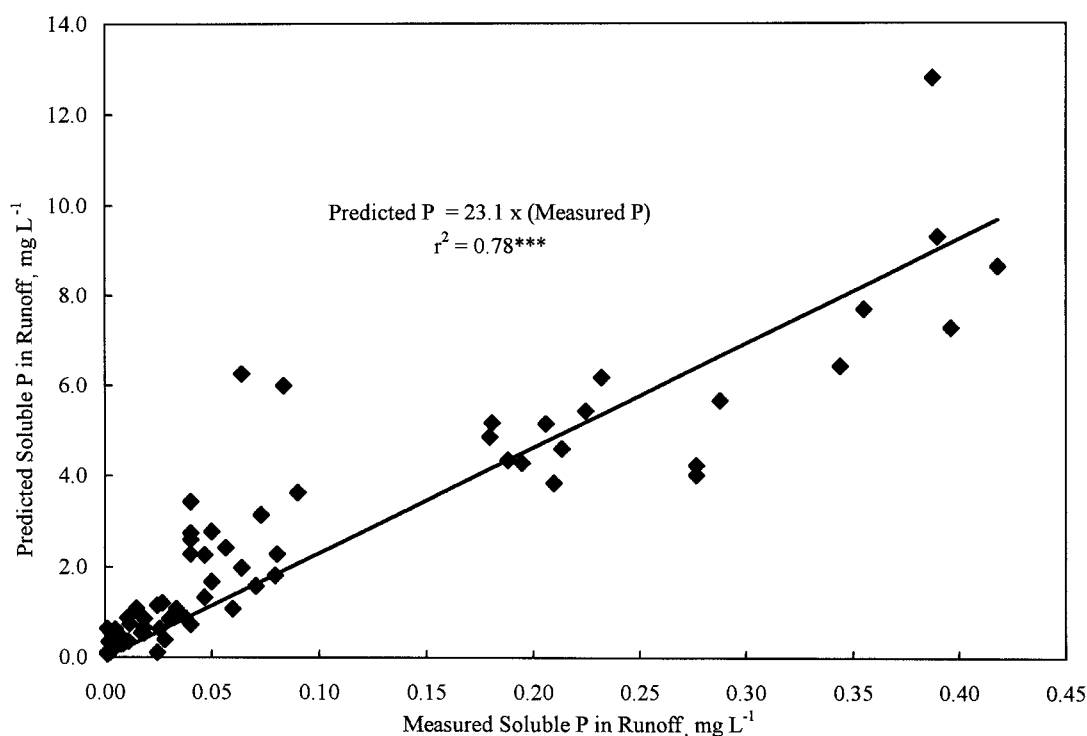


Fig. 1. Relationship between measured soluble P in runoff and predicted soluble P in runoff from the 78 soil boxes.

likely to be sources of significant overprediction. Because values for time of runoff (Eq. [2]), runoff amounts (Eq. [6]), sediment loss (Eq. [7]), and soil bulk density (Eq. [6]) were all measured, they also are not likely to be sources of significant overprediction. This essentially leaves the sources for error in values for degree of soil aggregation (Eq. [8]) and EDI (Eq. [6] and [7]). Our soils were given values for soil aggregation as measured by Sharpley (1985) for similar soil types, thus justifying the values we chose. A sensitivity analysis for our data showed that increasing values for soil aggregation by 50% would increase predicted values for soluble P in runoff by an average of 70% and would cause an average overprediction of soluble P in runoff of 34 times. Decreasing values for soil aggregation by 50% would decrease predicted values for soluble P in runoff by an average of 40%, but would still cause an average overprediction of soluble P in runoff of 12 times. This range of $\pm 50\%$ would cover all values for degree of soil aggregation measured by Sharpley (1985) for a wide range of soil types. Therefore, our values for degree of soil aggregation could have caused some of the overprediction of soluble P in runoff, but probably did not account for all of it. Also, because the work by Sharpley (1985) shows that the values that we did use for degree of soil aggregation were reasonable for our soil types, they were probably not a significant source of overprediction errors. The final source of overprediction error is EDI. Because sediment yield is used to calculate EDI in Eq. [7], this suggests that Eq. [7] may not be completely appropriate for the conditions of our experiments. However, the work by Sharpley (1985) in developing Eq. [7] and Sharpley and Smith (1989) in applying Eq. [7] suggests that Eq. [7] should improve the ability to predict P concentrations in runoff rather than cause overpredictions. This discussion essentially raises the point that there seemed to be no clear cause of overprediction of soluble P in runoff, given either a single variable or a combination of variables.

Another source of overprediction error may have been the way in which W (Eq. [6] and [11]) was calculated. The variable W represents the ratio of the amount of runoff water to the amount of soil with which it interacts to produce P desorption. For tillage conditions, calculating W as in Eq. [6] and [11] apparently leads to an overprediction of soluble P in runoff. We experimented with calculating W a few different ways to see the effect on overprediction errors in our data. If W was calculated as the ratio of runoff to sediment content in runoff, then predicted soluble P in runoff was on average 40% less than measured soluble P in runoff (data not shown). Essentially, considering W as the ratio of runoff to suspended sediment produces a system that is too dilute, where as considering W as in Eq. [6] and [11] produces a system that is too concentrated. If W was calculated as the average of these two extremes, then predicted soluble P in runoff and measured soluble P in runoff exhibited a 1:1 relationship (data not shown). This suggests that the amount of soluble P in runoff may simultaneously be a function of the interaction of runoff water with soil and the interaction of runoff water with the sediment suspended in that water. Therefore,

overprediction of runoff P in our results and the results of Daniel et al. (1993) may not be a function of the inability of Eq. [2] through [11] to estimate P desorption from soil to runoff water, but rather the inability of Eq. [2] through [11] to account for the reactions of the desorbed P and the suspended sediment in runoff water. Essentially, suspended sediment may be resorbing P out of the runoff solution and ultimately causing overprediction of soluble P concentrations in runoff. Sharpley et al. (1981b) conducted both field and laboratory experiments to determine if eroded sediment in runoff was likely to adsorb soluble P from runoff water. They collected runoff samples from 11 watersheds and observed that soluble P concentrations in runoff consistently decreased as sediment concentrations in runoff increased. In more controlled laboratory experiments using soil boxes and simulated rainfall, they determined that eroded sediment in runoff could indeed significantly adsorb P from runoff water. They concluded that eroded sediment should be considered as a P sink rather than a P source during a runoff event.

Daniel et al. (1993) presented all the data necessary to use Eq. [2] through [11] to predict soluble P concentrations in runoff from their pasture and tillage plots. In reanalyzing their data, we found that the degree to which soluble P in runoff was overpredicted, expressed as Predicted P/Measured P, was very well related to sediment yield measured during their runoff experiments (Fig. 2) as:

$$\text{Error} = (2.68) [\ln (\text{Sediment Yield})] + (4.94) \\ r^2 = 0.89 \quad [12]$$

Equation [12] was used with our data to adjust the predicted values of soluble P in runoff from simulated rainfall with the equation:

$$\text{Adjusted Predicted Soluble P} = \\ \frac{(\text{Original Predicted Soluble P})}{(\text{Error})} \quad [13]$$

The results are shown in Fig. 3. Adjusting the predicted soluble P in runoff data with Eq. [12] and [13] produced a nearly 1:1 relationship between measured and predicted values of soluble P in runoff. These results suggest that the source of overprediction errors in our data may indeed have been associated with sediment-runoff water interactions and the resorption of P by suspended sediment in runoff.

The discussion above also suggests that the way W is calculated and used in Eq. [2] through [11] to predict soluble in runoff should be reconsidered. However, because Sharpley and Smith (1989) and Daniel et al. (1993) successfully used Eq. [2] through [11] to predict soluble P in runoff for pasture situations, we maintain that the method used to calculate W is appropriate. Changing the way W is calculated would prevent successful prediction of soluble P in runoff for pasture situations. Therefore, we suggest that any overprediction of runoff P may be a function of the inability of Eq. [2] through [11] to account for resorption of runoff P by the suspended sediment in the runoff water, and that Eq. [12] may be used to account for P resorption. However, because

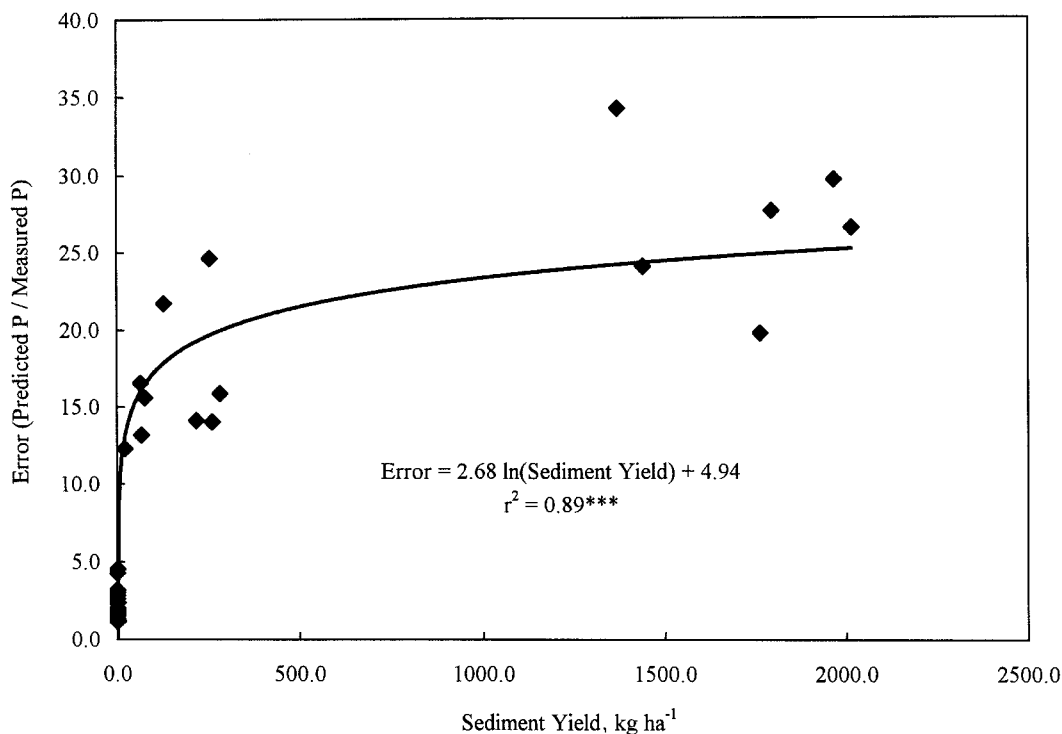


Fig. 2. Relationship between sediment yield and the error factor representing how much soluble P in runoff was overpredicted, as taken from the data of Daniel et al. (1993).

our experiments were not designed to investigate this resorption of P, Eq. [12] deserves further research.

CONCLUSIONS

In Delaware and throughout the Mid-Atlantic Coastal Plain, the FHANTM 2.0 model has the potential to estimate the risk of P loss from agricultural fields provided that its soil nutrient components accurately

represent field conditions. We have conducted research to determine if the P desorption equation $P_d = K P_o t^\alpha W^\beta$ can be incorporated into FHANTM 2.0 to estimate soluble P losses in runoff. Our results show that the equation overpredicts soluble P concentrations in runoff by a factor of about 20 for tillage type conditions where there is significant soil loss in runoff. However, adjusting predictions by a factor calculated from eroded sediment load in runoff resulted in a nearly 1:1 relationship be-

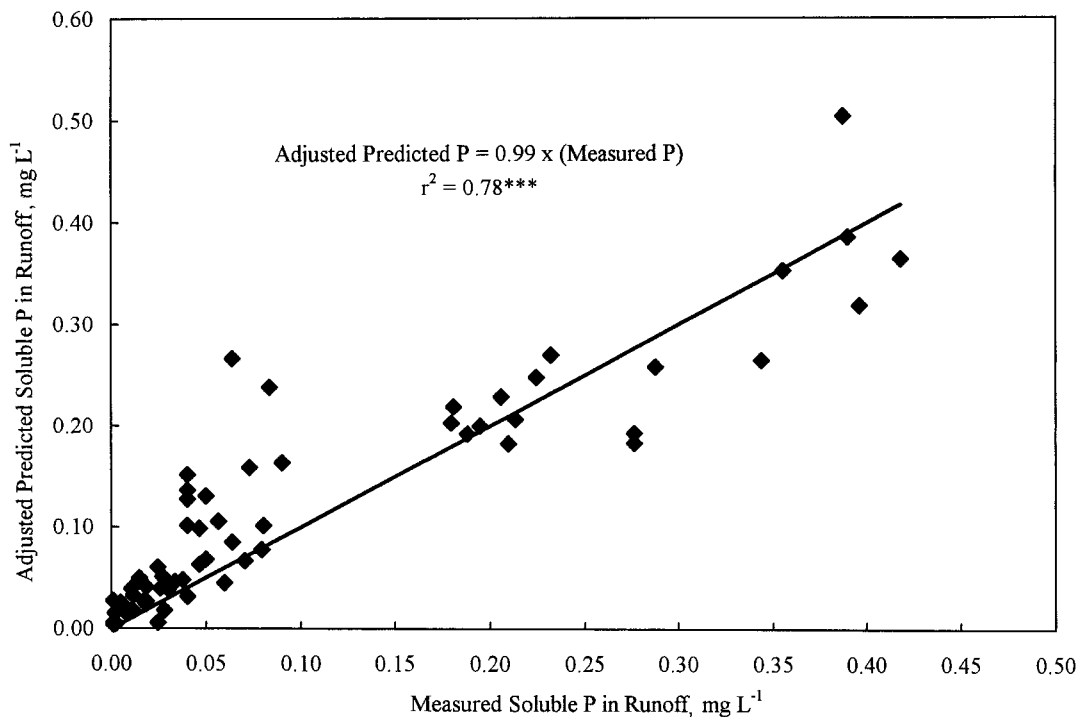


Fig. 3. Relationship between measured soluble P in runoff and predicted soluble P in runoff as adjusted based on EDI values for the 78 soil boxes.

tween measured and predicted runoff soluble P concentrations. The relationship between overprediction error and sediment loss in runoff suggests that P sediment may be resorbing P from the runoff solution, a phenomenon for which the P desorption equation does not account. However, because our experiments were not conducted to determine if sediment loss in runoff will reduce soluble P concentrations in that runoff, our data cannot confirm such a phenomenon. This point deserves future investigation.

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